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Restoration of wet meadows to enhance Gunnison sage-grouse habitat and drought resilience in arid rangelands

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The arid sagebrush landscape of the Gunnison Basin, Colorado is home to the federally threatened Gunnison sage-grouse (GUSG; *Centrocercus minimus*) and is expected to become hotter and drier with a changing climate. Wet meadows within the sagebrush ecosystem are a critical lifeline for wildlife and livestock, particularly during drought years, yet they occupy less than 2% of the landscape. Our objective was to enhance wet meadow drought resiliency by slowing the water down, reconnecting floodplains, and increasing wetland vegetation. Indirectly we also aimed to enhance GUSG habitat and improve rangeland condition. Between 2012 and 2020, we constructed nearly 900 low-tech restoration structures (Zeedyk structures), across seven drainageways with wet meadows. Six of these years were drought years. We used a before-after-control-impact design to assess vegetation response. Vegetation data were collected on 135 randomly selected treated and 30 control transects. We found that 75% of ephemeral units and all of perennial units achieved or surpassed the wetland plant cover management goal of a 4% yearly increase. This led to an average enhancement of 40% in wetland plant cover in the treated drainageways. There was a significant positive difference between treated and control transects in 50% of the drainageways, regardless of hydroperiod status. The low-tech restoration structures were effective at rewetting perennial and ephemeral wet meadows within the arid landscape, even during a megadrought, and reduced non-native invasive weeds in all but one of the treated units. Forbs and grasses critical to sage-grouse and important to livestock operations increased in 67% of the units.

Key words: climate-adaptation, sagebrush, vegetation monitoring, wetlands, Zeedyk structures

Implications for Practice

- Vegetation responses can serve as an indicator for measuring wet meadow restoration success.
- Changes in vegetation functional groups serve as a good proxy for drought resiliency and wildlife habitat condition.
- Restoration response rates vary across sites and hydroperiod type (ephemeral vs. perennial).
- Ephemeral meadows and drainageways, rather than perennial streams, dominate arid lands, and our results suggest that treating both types induce valuable changes.
- This simple climate adaptation strategy can increase drought resilience in arid lands.

Introduction

For over two decades, the southwestern region of the United States has been plagued by a prolonged drought, earning it the designation of a "megadrought." This current drought surpasses any other recorded megadrought dating back to at least the year 800 (Williams et al. 2022). Such droughts have significant consequences, impacting the quality of life for flora, fauna, and human populations by reducing water availability. Cook et al. (2015) and Wahl et al. (2022) have forecasted a rise

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in regional temperature stress and soil moisture depletion, which will lead to long recovery periods in the southwest United States by the mid-century. Additionally, between 2030 and 2059, the coterminous United States will experience a substantial decline in terrestrial water storage, as predicted by Pokhrel et al. (2021). The increasing frequency of droughts and reduced water storage underlines the necessity of climate change adaptation strategies to mitigate the detrimental impact on wildlife and humans alike.

Wetlands in the arid southwestern United States are a rare and precious resource, with a staggering 95% of the riparian habitats having been modified or destroyed by various land use practices over the past century (Krueper 1995). In such an arid region, these riparian habitats, which are critical to approximately 80% of all animal species and over 50% of breeding birds have been significantly impacted by historic land use (Carothers 1977; Krueper 1993). Vegetation within ephemeral and perennial stream channels is essential in preventing soil erosion, reducing floodwater velocity, and regulating temperature. Due to the increased availability of moisture, streams in arid and semiarid regions have higher levels of vegetation, wildlife diversity, and abundance than surrounding uplands (Levick et al. 2008). The abundance and diversity of wildlife species within a region are significantly influenced by the composition and variety of vegetation, and the destruction or harm to their habitats may cause a decline in breeding populations, as noted by Anderson et al. (1977).

A climate change vulnerability assessment for the Gunnison Basin in central Colorado was conducted in 2009 by The Nature Conservancy and others (Neely et al. 2011). Climatologists predicted that the climate would likely warm by 3°C and average summer temperatures would rise by approximately 4°C by 2050. The study identified the most vulnerable ecosystems in the basin as ephemeral drainages and perennial wetlands, particularly during periods of drought. In 2010, the Gunnison Climate Working Group (GCWG) was established, comprised of federal, state, educational, and private partners. The GCWG focused on developing an adaptation strategy for ephemeral drainages that include wet meadows and perennial springs that sustain the federally threatened *Centrocercus minimus* (Gunnison sage-grouse [GUSG]).

As stated in the GUSG Rangewide Conservation Plan (Gunnison Sage-grouse Rangewide Steering Committee 2005), as the upland sagebrush habitat dries out during summer, female GUSG and their broods choose riparian and wet meadow habitats that have plentiful forbs and moisture (Connelly & Markham 1983; Fischer et al. 1996; Donnelly et al. 2016). Peterson (1970) found that forbs are a primary food source for Greater Sage-grouse (*C. urophasianus*) chicks, accounting for up to 78% of their diet by volume. Another study conducted in Oregon by Drut et al. (1994) documented over 34 genera of plants, mostly forbs, and 41 families of invertebrates, primarily beetles, in the diet of Greater Sage-grouse chicks. Donnelly et al. (2016) found that population abundance of Greater Sage-grouse in other semiarid environments of Oregon, California, and Nevada increased with proximity to wet meadows and other mesic resources. According to Remington et al. (2021), the restoration of wet meadows and mesic resources and the promotion of resilience in the sagebrush ecosystem can aid in sage-grouse conservation efforts, particularly by improving nesting and brood-rearing habitats that could enhance chick survival and recruitment. This factor has been identified as a limitation in certain GUSG populations, as noted by Davis et al. (2015). Wet meadows also provide critical habitat for many non-listed species, such as neotropical migratory birds, amphibians, elk (*Cervus canadensis*), mule deer (*Odocoileus hemionus*), other wildlife, and domestic livestock. To mitigate the adverse impacts of droughts and flooding on wildlife and land-based livelihoods, the GCWG identified the restoration and enhancement of impaired wet meadows and drainageways as a crucial climate adaptation strategy.

The Wet Meadow Restoration and Resiliency-Building Project was initiated by the GCWG in 2012, employing low-tech restoration methods (Zeedyk & Clothier 2012; Wheaton et al. 2019) that have been demonstrated to enhance wetland vegetation and promote carbon sequestration by slowing the water down, mitigating erosive energy, and encouraging late-season water flows (Silverman et al. 2018; Norman et al. 2022).

Our objective was to evaluate the effectiveness of low-tech, process-based restoration techniques in improving drought resiliency and enhancing habitat quality for GUSG. To achieve this, we assessed the vegetation response on seven treated drainageways, consisting of eight ephemeral and four perennial units. Our analysis was based on plant species composition, with the following questions guiding our evaluation: (1) Did the treatment meet our management objective? (2) Was there a discernible difference in vegetation response between treated and control areas? (3) Was there variation in response among functional groups (forbs, grasses, and rushes)? (4) Did ephemeral units exhibit a different response than perennial units? (5) Did non-native species exhibit different responses than native species? and (6) Which individual species exhibited the greatest rate of change in cover?

Methods

Study Area Description

The sagebrush ecosystem in the Upper Gunnison River Basin covers an area of approximately 202,000 ha. The elevation ranges from 2,200 to 2,900 m. Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*) dominate the lower and drier elevations and mountain big sagebrush (*Artemisia tridentata* spp. *vaseyanna*) dominate the upper and wetter elevations. Drainageways associated with perennial springs and wet meadows occupy less than 2% of the sagebrush landscape. Impacts from historic land use, e.g. wagon tracks, roads, trails, stock ponds, livestock grazing, etc., have caused erosion and channelization, thus reducing the cover of wetland vegetation. The landscape is primarily managed by the U.S. Forest Service and Bureau of Land Management, with only 20% being privately owned and managed. The livelihoods of the people who depend on the land, such as livestock grazing and hunting, rely heavily on wetlands, which provide important ecosystem services on both public and private lands (Knapp 2011).

Climate

The climate in Gunnison, Colorado from the 1971 to 2000 thirty-year average (Colorado Climate Center 2020; http://climate.colostate.edu/data_access.html) is best characterized as cold winters and warm wet summers. Winter days average a low of −19.1°C and an average high of −1.4°C. Summers experience an average high of 25.7°C and low of 4.3°C. Annual precipitation averages 230 mm but is highly variable (29–360 mm). Winter precipitation averages 58 mm, in the form of snow, and summer averages 90 mm. March (13 mm) and June (15 mm) are the driest months; wettest months are July and August (33 and 42 mm, respectively). During the course of this study (2012–2020), average annual temperature was 0.3°C warmer than 1971–2000 average. Droughts during 2012–2020 were more common than wet periods; 6 of 9 years were considered a drought; 2014–2015 and 2019 were the only wet years (Fig. 1; Abatzoglou et al. 2017).

Study Design and Field Methods

Within the semiarid sagebrush zone of the Gunnison Basin wet meadows often form in linear drainageways where surface and groundwater flow concentrates (sensu "vegetated drainageway" in BLM 2023). Wet meadows and their associated drainageways were selected as candidates for restoration through a Geographic Information System analysis that took into account proximity to GUSG breeding sites (within 3.2 km from the wet meadow), sagebrush and wet meadow conditions, local knowledge, and accessibility (data not published). Final drainageway selection was based on field observations, which considered the degree of degradation, accessibility for treatment construction, and GUSG sign, such as cecal and fecal droppings. Restoration ecologist Bill Zeedyk designed low-tech and mostly hand-built structures, such as one-rock-dams, Zuni bowls, and media lunas, to slow the water down, reduce erosion, re-connect the channel to the floodplain, stabilize head cuts, raise the water table, and increase wetland plant cover (Zeedyk 2009; Zeedyk & Clothier 2012). Volunteers and youth conservation crews helped build nearly 900 structures. The restoration effort took place across private and public lands within designated critical habitat for GUSG (U.S. Fish and Wildlife Service 2014).

Between 2012 and 2020, the GCWG treated 11 headwater drainageways and monitored nine of these. Our analysis includes the seven drainageways that had at least 5 years of post-treatment monitoring (Table 1; Fig. S1). Watershed area of the seven drainageways ranged from 2.5 to 12.7 km^2. Stream channels and associated wetland vegetation ranged from 5 to 75 m wide (Table S1). Within a drainageway, segments or reaches could alternate between ephemeral and perennial based on the presence of springs or other water inputs. Ephemeral segments did not typically retain surface water for extended periods. The wetland vegetation in these areas was mainly composed of facultative wetland species such as Baltic rush (*Juncus arcticus*) and rarely included obligate wetland species. Perennial units, on the other hand, had surface water flowing during the growing season, usually from springs, and saturated soils that were dominated by obligate wetland species like beaked sedge (*Carex utriculata*). Many, but not all drainageways, included both ephemeral and perennial segments.

Our design was a before-after-control-impact study. The before–after aspect of the study assessed whether we met our management objective, while the control-impact component compared the plant species composition of treated versus untreated (control) areas over time. Our management objective was to increase the average cover of obligate and facultative wetland plant species, including sedges, rushes, and forbs, in the treated areas by a minimum of 20% within 5 years following treatment.

Vegetation data were obtained from permanent transects that were randomly placed in treated areas (135) and control areas (30), the latter of which were located in the same drainageway as the treated areas but typically up-valley. Permanent transects were established on approximately 15% of the structures within a treated drainageway, using a stratified random sample design, and a random start was selected within the first set of structures. If our random sampling method did not choose at least one of each type of restoration structure, we manually selected one of each type. Transects were generally placed above restoration structures, except for

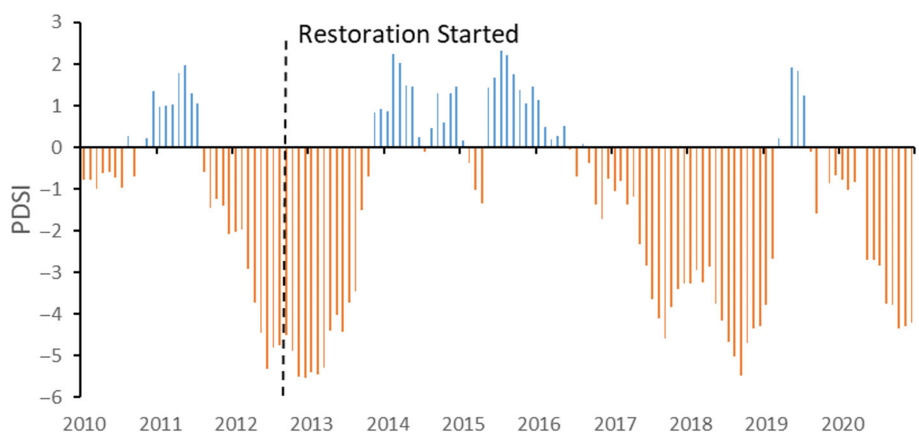

Figure 1. The Palmer Drought Severity Index (PDSI) for Gunnison County, Colorado is reported on a monthly basis from 2010 to 2020. Blue bars represent wet months and orange bars represent drought months. https://droughtindexportal.colorado.edu/

Table 1. The average annual percentage change for wetland plant cover, total plant cover, bare ground, forbs, rushes, and grasses cover is presented for each unit. The calculation of the mean percent change per year was based on the predicted initial and final cover, as determined by linear regression. Refer to Table S1 for additional attributes. *Management goal was a gain of 20% in wetland plant cover over 5 years (or an average of 4%/year). Green highlighting = goal met; orange highlighting = goal not met; red font indicates an undesired result.

			Mean % change/year					
Unit code	*No. of trt years*	*Grazing use*	*Wetland plants**	*Plant cover*	*Bare ground*	*Forbs*	*Rushes*	*Grasses*
Ephemeral								
RDN	8	Moderate	59	15	−10	13	438	6
SH	5	Light	15	2	−5	2	18	−1
WLF	8	Light	7	6	−7	4	7	6
CTT	5	Heavy	8	1	2	7	9	−1
CHN	6	Light	10	10	−15	21	10	9
FTH	7	Heavy	5	2	8	−3	5	4
FTAE	6	Heavy	−2	−3	14	−1	−1	−5
FTE	7	Light	−6	−4	21	−8	−7	1
Perennial								
SH-P	5	None	7	0	100	2	8	−4
KZR-P	6	Heavy	6	4	13	7	14	2
WLF-P	7	Light	6	7	−12	3	2	7
CHN-P	6	None	7	4	−11	0	6	5

media lunas and plug and spreads, which were placed below, to correspond with the areas that were expected to respond to the treatment structure. Transects crossed the stream channel and associated floodplain and ran from upland edge to upland edge, resulting in variable lengths (5–75 m). We classified each transect independently as ephemeral or perennial based on hydroperiod and presence or absence of obligate wetland species.

We collected live or dormant foliar cover data every 0.5 m along a meter tape using the line-point-intercept method (Canefield 1941), noting all the species at each point. Total cover on a transect could exceed 100% due to overlapping layers in the canopy. Bare soil, rock, or detached litter was only recorded if live vegetation was not encountered. Percent cover by species was calculated for each transect. Data were collected annually during the same Julian period as baseline data collection. We took photographs from the 0-m mark and end of each transect, with the transect line appearing vertically in the middle of the photo, and looking upstream, with the transect tape appearing horizontally in the photo, to enable visual comparisons from year-to-year. We also measured the tallest vegetation height at every meter along the transects, rather than the average stubble height. Grazing utilization was not a restoration criterion, but we used Anderson and Currier's (1973) categories and definitions (none, light, moderate, and heavy) to determine relative utilization based on field observations. Finally, we noted that the grazing regime was consistent over the study period, except for RDN, where livestock grazing pressure was reduced by the landowner after the structures were built. Drainageway characteristics are summarized in Table S1.

Data Analysis

The sampling unit was a transect. However, for ease of discussion, results were generally summarized by pooling all transects that share both a hydroperiod and a drainageway, hereafter called "units." For example, the ephemeral transects in Sage Hen drainageway constitute one summary unit, regardless of whether or not they were located in a contiguous segment. In total, there were 12 hydroperiod–drainageway combinations or units, eight ephemeral and four perennials. In order to evaluate whether the management objective was achieved (as per the before–after study design), we constructed linear regression models for treated transects and all functional groups. By using the regression equations, we calculated the linearized cover for the first year (prior to treatment) and the final year (2020). We then calculated the relative change (%) in cover between these time points, as well as the mean annual change.

For the control-impact study design, we assessed whether the treatment transects were statistically different from the untreated areas by determining trends based on the slopes in treated and control transects. We conducted a normality test on the slopes and found them to be normal. To test for significance at an alpha level of 0.05, we used an unpaired t test that is robust to unequal sample sizes. The results are presented separately for ephemeral versus perennial transects.

We used ordinary least-squares correlation to investigate if there was a relationship between the starting condition and the amount of improvement. Specifically, we summarized the slope (% change/year) of ephemeral transects within each drainageway on the y-axis and its corresponding pre-treatment cover (%) on the x-axis.

We classified plant species into six functional groups: (1) wetland species (including obligate and facultative wetland), (2) forbs, (3) sedges, (4) rushes, (5) grasses, and (6) native/non-native (USDA, NRCS 2020). To qualify as a wetland species, a plant had to be classified as either an obligate or facultative wetland species. Obligate wetland species were those that occurred in wetlands under natural conditions with a probability of over 99%. Facultative wetland species were those that were usually found in wetlands (with a probability of 67–99%) but could also occur in non-wetland areas. Reported values are means unless stated otherwise, followed by the corresponding range of values in parentheses.

Results

A total of 215 plant species were identified (Table S2), with 25 being common and distributed throughout the range. The cover data allowed us to estimate biomass, while the trend data enabled us to assess changes over time. We categorized the results based on functional groups, including wetland plants, forbs, grasses, rushes, and non-natives (Table S2).

Before–After Treatment and Management Objective

We successfully achieved our management objective of increasing wetland plant cover by 20% over 5 years, post-treatment, in 75% of the ephemeral units (six out of eight) and all four perennial units (Table 1; Figs. 2 & 3).

The wetland plant cover in ephemeral units exhibited an average annual gain of 12% (−6 to 59%; Table 1). All units, except for FTAE and FTE, met the management objective (Tables 1 & S3). The pre-treatment wetland plant cover in ephemeral units averaged 47% (17–72%), and by the final year, cover averaged 66% (37–101%), resulting in an overall increase of 40% of pre-treatment cover (Table S3; Figs. 2 & 3).

The wetland plant cover in perennial units showed an annual gain of 6% (6–7%; Tables 1 & S3). The pre-treatment wetland plant cover was 90% (65–104%), and by the final year, the cover was 125% (87–145%), showing an overall increase of 39%.

To understand the variability in ephemeral units, we examined the relationship between pre-treatment cover and percent change. A negative correlation was found between pre-treatment cover (%) and average % change/year for wetland plant cover in ephemeral units ($r^2 = 0.53$, $p = 0.02$; Fig. 4). Ephemeral units with lower initial cover, where more gains could be made, showed higher annual improvements while units with over 65% wetland plant cover did not improve or experienced loss.

Total Plant Cover. Biomass estimation can be inferred from total plant cover, regardless of the plant species. The total plant cover in ephemeral units exhibited an annual gain of 5% (−4 to 15%). All units, except for FTAE and FTE, gained cover (Tables 1 & S4). The pre-treatment total plant cover in ephemeral units was 132% (82–189%), and by the final year, cover was 158% (111–231%), resulting in an overall increase of 20% (Table S4; Fig. 2). The total plant cover in perennial units showed an average annual gain of 2% (0–6%; Tables 1 & S4). The pre-treatment total plant cover was 180% (154–211%), and by the final year, the cover was 221% (184–264%), showing an overall increase of 23% of pre-treatment cover. Despite the absence of a significant correlation ($r^2 = 0.37$, $p < 0.1$; Fig. 4) between pre-treatment total plant cover and the yearly change

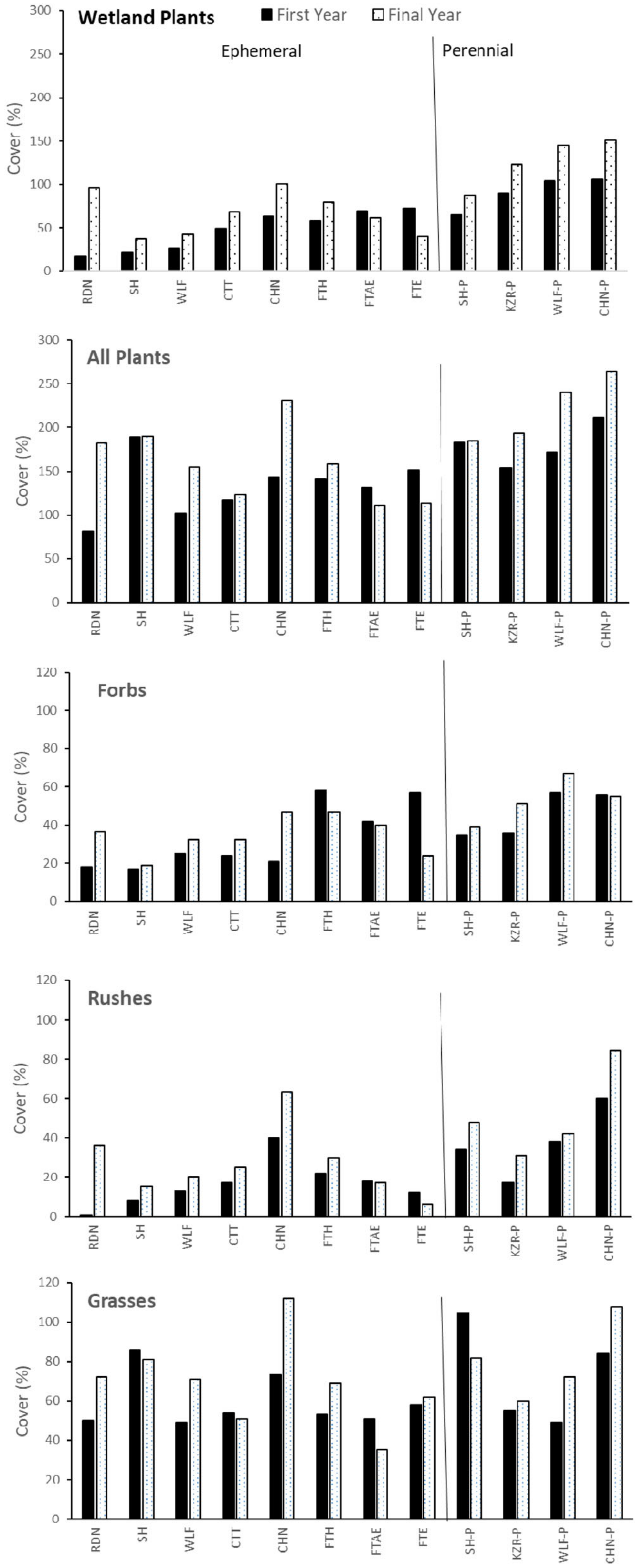

Figure 2. First and final year cover (%) for functional groups (wetland species, all plants, forbs, rushes, and grasses). To adjust for the variability in annual precipitation, a linear regression equation was employed to estimate the first year (pre-treatment) and final year cover percentages (see Table S3 for number of treated years).

Figure 3. Repeat photos demonstrate the increase in wetland cover associated with a media luna structure (top row). The lower row demonstrates a log and fabric structure used to heal a head cut. Note that an additional log structure was added after the 2013 photo was taken. The white ribbon visible in the lower photos marks one vegetation transect; note the depth of the stream incision and sediment build up below the tape in both photos. All photos were taken within a few Julian weeks of the original photo.

with treatment in ephemeral units, the general trend was similar to wetland plant cover.

Forb Cover. Forb cover is an important indicator of GUSG habitat. The forb cover in ephemeral units exhibited an annual gain of 4% (−8 to 21%). All units (67%), except for FTH, FTAE, and FTE, gained cover (Tables 1 & S5). The pre-treatment forb cover in ephemeral units was 33% (17–58%), and by the final year, cover was 35% (19–47%), resulting in an overall increase of 6% (Table S5; Fig. 2).

The forb cover in perennial units showed an annual gain of 3% (0–7%; Tables 1 & S5). The pre-treatment fob cover was 46% (35–57%), and by the final year, the cover was 53% (39–67%), showing an overall increase of 15%. There was a significant correlation between pre-treatment cover and annual change in cover for ephemeral units ($r^2 = 0.58$, $p = 0.01$; Fig. 4). Sites that had greater than 40% forb cover in the pre-treatment condition failed to improve, whereas sites that started with less than 40% cover increased over time.

Rush Cover. Baltic rush, a wetland plant that can tolerate different moisture levels, dominated the rushes. The rush cover in ephemeral units exhibited an annual gain of 60% (−7 to 438%). All units, except for FTAE and FTE, gained cover (Tables 1 & S6). The pre-treatment rush cover in ephemeral units was 16% (1–40%), and by the final year, cover was 27% (6–63%), resulting in an overall increase of 62% (Table S6; Fig. 2). The rush cover in perennial units showed an annual gain of 7% (2–14%; Tables 1 & S6). The pre-treatment rush cover was 38% (17–60%), and by the final year, the cover was 51% (31–84%), showing an overall increase of 34%. There was no significant correlation between pre-treatment cover and annual change in cover with treatment ($r^2 = 0.2$) in ephemeral units.

Grass Cover. Grass cover is a vital indicator for herbivores such as elk and cattle as well as grasshoppers, which are a key food source for sage-grouse. The grass cover in ephemeral units exhibited an annual gain of 1% (−5 to 9%). All units (67%), except for SH, CTT, and FTAE, gained cover (Tables 1 & S7). The pre-treatment grass cover in ephemeral units was 59% (49–86%), and by the final year, cover was 69% (35–112%), resulting in an overall increase of 17% (Table S7; Fig. 2). The grass cover in perennial units showed an annual gain of 6% (2–14%; Tables 1 & S7). The pre-treatment grass cover was 73% (49–105%), and by the final year, the cover was 81% (72–108%), showing an overall increase of 10%. There was no

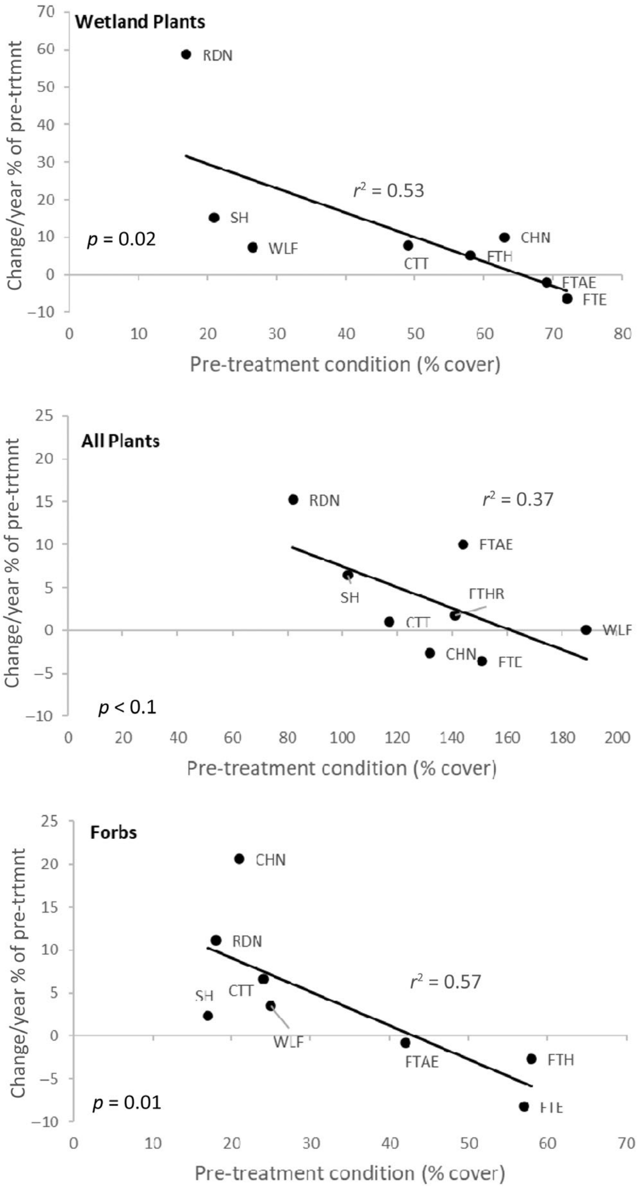

Figure 4. A comparison between the pre-treatment condition (% cover) and the annual change following treatment. To determine if the amount of improvement had a relationship to the starting condition, we used ordinary least-squares correlation, in which each units' slope (% change/year) is displayed on the y-axis and its corresponding pre-treatment cover (%) is given on the x-axis.

significant correlation between pre-treatment cover and annual change in cover. Although we did not analyze the effects of livestock grazing, we have provided relevant data in Tables 1 & S1.

Bare Ground. Measurements of bare ground are a useful indicator for evaluating treatment effects in ephemeral units but are typically less relevant for perennial units, where bare ground is less common. Among the ephemeral units, the percentage of bare ground decreased (a desirable outcome) in 50% of the sites, but notable increases were observed in three units (FTH, FTAE,

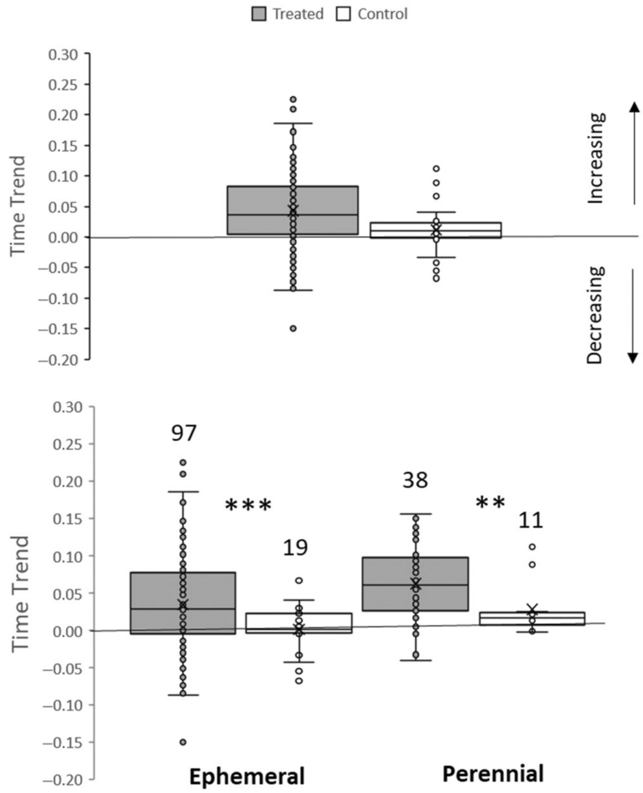

Figure 5. Time trend (slope) for treated and control transects. A slope was calculated from wetland species cover data for each transect for all sample years. A positive time trend indicates an increase in cover, whereas a negative time trend (<0) indicates a decrease. In this and subsequent box plots, individual transect values shown by small circles; means by ×. Boxes represent the median and interquartile ranges (IQR: 25th–75th percentile). Range lines extend to cover all points within 1.5 IQR of median; more distant points are outliers, shown as small circles beyond the furthest horizontal lines. t Tests, assuming unequal sample size was conducted for each treatment–control comparison. Upper graph compares all treated transects to control transects regardless of hydro-type. Lower graph contrasts treatment and control transects within ephemeral and perennial units. *p less than 0.05, **p less than 0.01, ***p less than 0.001.

and FTE; Table S8). It should be noted that initial increases in bare ground can result from sediment deposition caused by the structures, which slow down water velocities. Wetland plants generally begin to establish themselves in areas with sediment buildup within 1–3 years, depending on soil moisture levels.

Control-Impact

Wetland Plants. Wetland plants significantly gained cover in treated transects compared to the controls ($p < 0.001$; Fig. 5). Generally, perennial transects had steeper increases (higher rate of change) than ephemeral transects (Fig. 5). Wetland cover decreased in a few transects, including 23% of ephemeral transects compared to 11% of perennial transects. We reviewed the pattern among drainageways to determine if the loss was range-wide or specific to certain drainageways. We noted that 65% of the transects that exhibited a loss were associated with the Flat Top drainageway and the remaining 35% were scattered

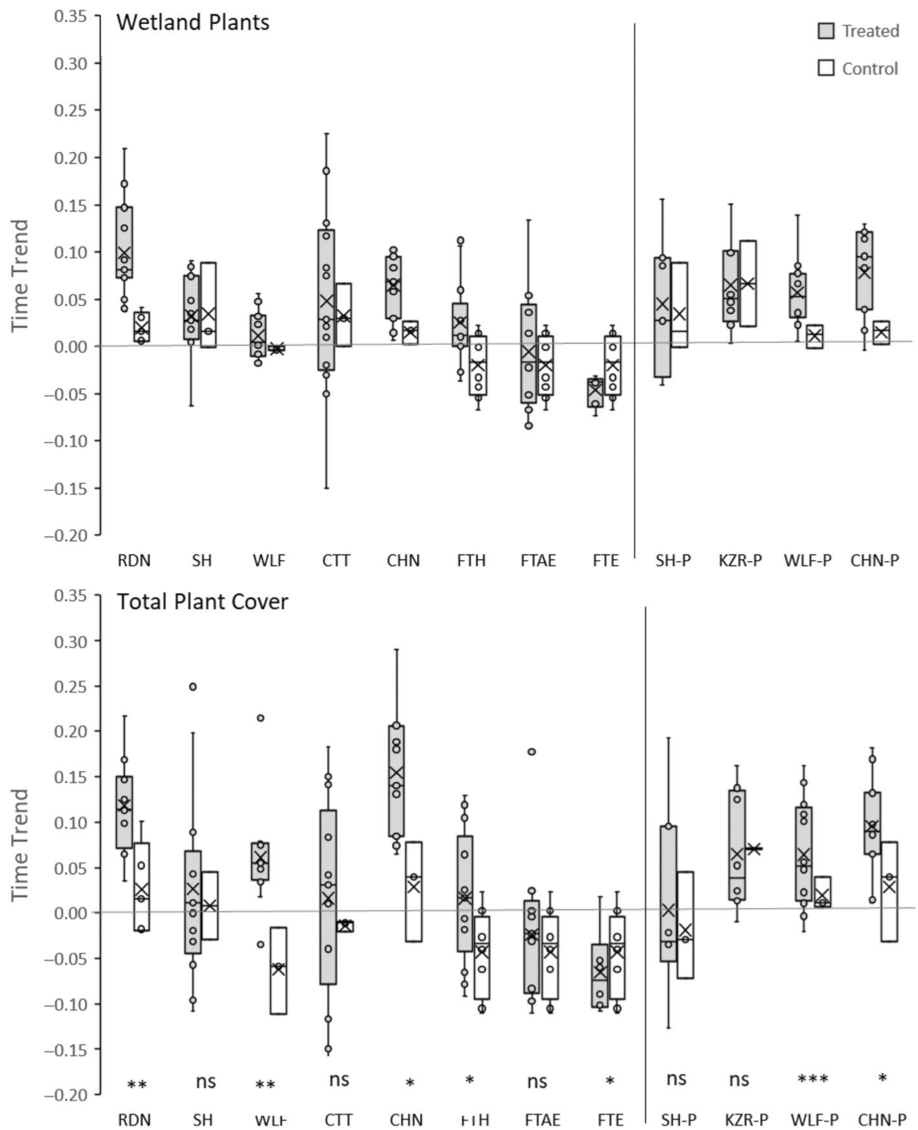

Figure 6. Time trend (slope) for wetland plants and total plant cover, comparing treated to control transects for each unit. Other conventions as in Figure 5.

among two other drainageways, WLF and CTT. At the unit level, 50% of the ephemeral and perennial units had significant gains in wetland plants compared to the controls, one unit (FTE), had a significant loss, and change was not significant for the remaining five (Fig. 6).

Total Plant Cover. In half (50%) of the treated ephemeral units, there was a significant increase in total plant cover compared to the controls, whereas a decline was noted in one of the eight (12.5%, Fig. 6). Similarly, in half of the perennial sites, there was a significant increase in total plant cover compared to the controls. The other half of the units was not significant but showed trends in an improving direction (Fig. 6).

Forb Cover. In 38% of the treated ephemeral units, forb cover significantly increased, whereas in two other units, we observed a substantial but not statistically significant difference in the rate of change between treated and control transects. Forb cover significantly decreased in 25% of the treated units (two of eight) compared to the controls (FTH and FTE). Among perennial units, only KZR showed a significant gain in forb cover in treated transects compared to the controls, but all other units had the expected pattern, that is an overall gain in treated and overall losses in the controls (Fig. 7).

Grasses and Rushes. In 37% of the treated ephemeral units, grass cover showed a significant increase compared to the controls (Fig. 7), whereas none of the perennial units showed a significant difference (Fig. 7). For rushes, there were significant differences between treated and control transects in 50% of the ephemeral units (Fig. 7), and only one site (25%) of the perennial units showed a significant difference (Fig. 7).

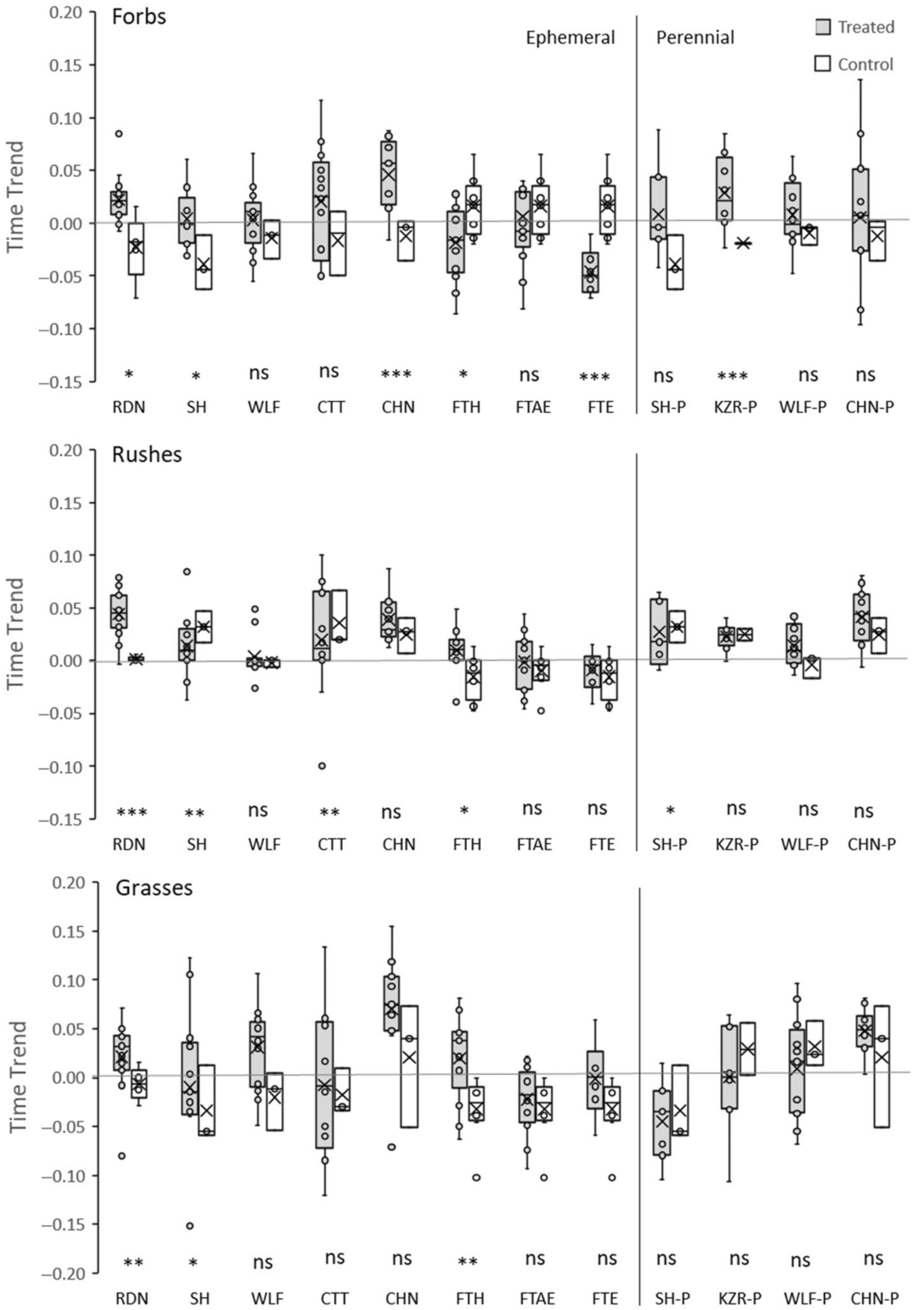

Figure 7. Time trend (slope) for forbs and grass cover, comparing treated to control transects for each unit. Other conventions as in Figure 5.

Species With the Greatest Rate of Change

Refer to Table S2 for a list of plants, including scientific and common names along with their codes and functional group categories.

Wetland Plants. Among the 215 plant species identified (Table S2), 13 wetland species were frequent and common, and all but two had a positive rate of change. Baltic rush and western aster (*Symphyotrichum ascendens*) were the most abundant, regardless of hydroperiod (Fig. S2). Obligate wetland species woolly sedge (*Carex pellita*), silver weed (*Argentina anserina*), Nebraska sedge (*Carex nebrascensis*) and beaked sedge occurred only in perennial units. Baltic rush had the greatest rate of positive change, whereas western aster had a greater positive rate of change in perennial units and essentially no average change in ephemeral units due to three units exhibiting a decline. RDN was the most notable unit where Baltic rush had an annual increase of 59% (Table 1) with a final cover of 36%, compared to a pre-treatment cover of 1% (Table S6).

Forbs. Six forb species were frequently observed across the study area (Fig. S3). The western aster and common yarrow

(*Achillea lanulosa*) were the most abundant among them. Western aster, wild iris (*Iris missouriensis*), and silverweed cinquefoil (*A. anserina*) showed an increase in their cover, while dandelion (*Taraxacum officinale*) and Canada thistle (*Cirsium arvense*) experienced a decline (Fig. S3).

Grasses. The two most abundant grasses were Kentucky bluegrass (*Poa pratensis*) and western wheatgrass (*Pascopyrum smithii*). Kentucky bluegrass, which is highly palatable to cattle, was abundant in all sites, with a cover ranging 32–64% in perennial units and 20–49% in ephemeral units (Fig. S2). Grass cover in the perennial units did not change much over time, but in the ephemeral units, as soil moisture increased, grass cover also increased. Western wheatgrass was present in all but one ephemeral unit and two of four perennial units, and the time trend was mixed (Fig. S3).

Non-Native Plants. A total of 24 species were categorized as non-native, with eight being widespread or invasive (Fig. S3), according to USDA, NRCS (2020) and Smith et al. (2020). Among the non-native species, only Kentucky bluegrass had high cover (Fig. S2). Dandelion, which is preferred as forage for sage-grouse chicks, had low cover, rarely exceeding 3% in perennial units and ranging 1–8% in ephemeral units. Most units showed a slight decline in non-native invasive species (Fig. S2). Smooth brome (*Bromus inermis*), an aggressive grass that competes with native grasses and does not benefit GUSG, showed mixed response to treatment. It increased in cover in CHN in both perennial and ephemeral units but decreased in WLF-P. Canada thistle was mostly absent in ephemeral units but present in all perennial units (2–9% cover). It decreased in all treated perennial units, with KZR-P and WLF-P experiencing the highest reduction, from 11 and 16% pre-treatment cover to 1 and 3%, respectively, representing an over 80% reduction.

Discussion

In seven different headwater drainages that included wet meadows surrounded by arid sagebrush uplands, we constructed approximately 900 structures. These structures were developed to slow the water flow, control erosion, spread the water out, and restore former wetlands. From 2012 to 2020, 6 of 9 years experienced drought conditions. The initial sampling year, prior to treatment, varied by site. Restoration and monitoring of certain sites began during a drought year, while others commenced during a wet year. To reduce the influence of individual years we assessed only the overall linear trend. Despite a 20-year drought that persisted (Abatzoglou et al. 2017; Williams et al. 2022), the treated areas experienced a general increase in wetland plant cover. During the course of the study, wetland vegetation coverage increased by an average of 40%, from an average of 62% coverage preceding treatment to 86% coverage in the final year (2020).

Our management objective was to increase wetland plant cover by an average of 4% per year within 5 years of treatment. We successfully achieved this objective in 10 of the 12 units. The remaining two units, FTAE and FTE, showed negative responses to treatment (−2 and −6%, respectively) and require further investigation. Despite being ephemeral units, they did not exhibit the highest rate of improvement, as observed in other similar units such as CTT. Grazing pressure did not seem to be the primary reason for the negative response, as CTT also experienced heavy grazing but still exceeded our management objective. Our design was not intended to address grazing intensity, so it is possible that grazing intensity is a factor. Pre-treatment wetland plant cover was found to be a good indicator of annual change ($r^2 = 0.53$), as sites with low wetland plant cover experienced the greatest increases. FTAE and FTE were among the wettest of the eight ephemeral units, suggesting that they had the least potential for improvement. Future studies will explore other variables that possibly contribute to the observed variability in wetland plant cover, such as watershed area, geology, elevation, hydrology, proximity to springs, aspect, grazing utilization, and time, which may be responsible for the negative response in FTAE and FTE.

The control-impact study revealed a significant difference in wetland plant cover trends between the treated and control transects ($p < 0.0001$). When examining the treated units individually, half of them displayed an increase in wetland plant cover compared to their respective control transects. However, FTE showed lower wetland plant cover than its corresponding control area. Finding suitable control areas within the same drainage proved challenging, as structures were built on one of the original control transects in KZR and CHN, leading to a reduced sample size for control transects. Nevertheless, controls in RDN, WLF, SH, CTT, FTH, FTAE, and FTE were considered appropriate.

Silverman et al. (2018) utilized normalized difference vegetation index (NDVI), which is an index of vegetation photosynthesis, to conduct a remote sensing analysis of restoration sites in Oregon, Nevada, and Colorado both before and after restoration. They analyzed the Gunnison Basin sites that were monitored for this study albeit with a shorter period (1–5 years) post-treatment. The researchers combined all treated units and compared them to off-site control units, and while they did not report on individual units, their results indicated a net increase in vegetation productivity in the treated areas compared to similar controls. A separate NDVI analysis on the Gunnison Basin drainageways is currently being prepared (T. Chapman 2023, The Nature Conservancy, Boulder, CO, U.S.A., personal communication).

We aimed to restore wet meadows to benefit the federally threatened GUSG, which depends on wet meadows for food during drought years when palatable forbs and insects are scarce in sagebrush uplands. Sage-grouse chicks feed mostly on forbs (78%) and insects (17%) during late summer (Peterson 1970). Our wet meadow treatments resulted in an average increase in forb cover of 4%/year. Ephemeral units showed a range of −8 to 21%, whereas perennial units ranged from 0 to 7%/year. While we did not measure insect abundance and diversity, previous studies have shown a positive relationship between plant biomass and insect abundance (Siemann 1998; Haddad et al. 2001). Prather et al. (2020) reported that arthropod abundance increased with soil moisture, regardless of plant diversity.

Whiles and Goldowitz (2005) found that ephemeral sites had the highest invertebrate taxon richness and diversity across a hydrologic gradient from ephemeral to perennial wetlands.

In some cases, increased soil moisture can lead to an increase in invasive species. For example, Wilson and Norman (2022) found that non-native species increased in two restored wetlands in Arizona. However, in our study, non-native species accounted for less than 1% of the total species and only one was dominant. The highly palatable Kentucky bluegrass was present in all sites, with cover ranging 32–64% in perennial units and 20–49% in ephemeral units. There was little change in the perennial units, whereas in ephemeral units, as soil moisture increased, the cover of Kentucky bluegrass also increased.

Our treatments had a mixed impact on the aggressive grass smooth brome, which does not benefit GUSG. It is possible that saturation negatively affected its growth, but until then, smooth brome tends to thrive with increased soil moisture. On the other hand, Canada thistle, the only state-listed noxious weed encountered on monitoring transects, showed a decrease in cover in all treated areas except for one site (SH). The two sites with the highest initial Canada thistle cover were in perennial units. KZR and WLF, experienced a significant reduction (11–1% and 16–3%, respectively), resulting in an overall loss of over 80%.

Of the 215 plant species we recorded in our transects, only four had an average cover exceeding 10%. The facultative wetland species, Baltic rush and western aster, showed the highest increase in cover, whereas upland species like western wheatgrass and Kentucky bluegrass had smaller increases. Baltic rush has a deep rooting depth (up to 60 cm) and can withstand wetland degradation, making it a reliable indicator plant for restoration monitoring as it responds quickly to rewetting (Stasiak 1994).

Ephemeral and intermittent streams play a crucial role in groundwater recharge in arid and semiarid regions by facilitating channel infiltration and transmission losses (Levick et al. 2008). Ephemeral and intermittent streams constitute approximately 81% of all streams in the arid and semiarid southwest (Arizona, New Mexico, Nevada, Utah, Colorado, and California), according to the U.S. Geological Survey National Hydrography Dataset (Levick et al. 2008). They offer a variety of ecological benefits such as forage, cover, nesting, water, and movement corridors for wildlife and livestock. Any changes in plant community abundance or composition are likely to have an impact on a variety of ecosystem functions and processes.

Our study indicates that a low-tech restoration approach is effective for both ephemeral and perennial units in the Gunnison Basin, Colorado. Perennial units showed less variability in vegetation response, ephemeral units demonstrated a wider range. This variability allows for a future comprehensive approach, incorporating the interplay of ecology, hydrology, and geomorphology (Zeedyk & Clothier 2012; Meitzen et al. 2013; Chambers et al. 2021). Using an ordination analysis, one can identify patterns that may inform hypotheses about the factors influencing response rates and could help identify areas where treatment is likely to be most effective. However, we recognize that our treated sites are still undergoing significant change 5–8 years after restoration, and that ongoing monitoring is necessary to fully understand the long-term impacts of these interventions. Additionally, structures used for restoration may continue to affect the hydrology of a water course even after sediment has filled them, further highlighting the need for continued monitoring.

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Supporting Information

The following information may be found in the online version of this article:

Figure S1. Map of restoration areas in the Gunnison Basin, Colorado, U.S.A.
Figure S2. The most common and wide-ranging wetland species cover and time trend.
Figure S3. The most common and wide-ranging forb and grass species cover and their time trend.
Table S1. Sampling effort and descriptive attributes by reach.
Table S2. Plant list from monitored sites.
Table S3. Wetland cover from pre-treatment year (first year) and final year (2020), total change (%), and average annual change for all reaches.
Table S4. Total plant cover from pre-treatment year (first year) and final year (2020), total change (%), and average annual change for all reaches.
Table S5. Total forb cover from pre-treatment year (first year) and final year (2020), total change (%), and average annual change for all reaches.
Table S6. Total rush cover from pre-treatment year (first year) and final year (2020), total change (%), and average annual change for all reaches.
Table S7. Total grass cover from pre-treatment year (first year) and final year (2020), total change (%), and average annual change for all reaches.
Table S8. Bare ground, including bare soil, litter, or rock (only recorded if vegetation was not present) from pre-treatment year (first year) and final year (2020), total change (%), and average annual change for all reaches.

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